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Integrated assessment of management strategies for metal-contaminated dredged sediments – What are the best approaches for ports, marinas and waterways?

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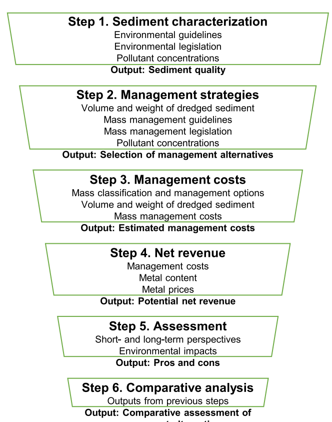
HIGHLIGHTS

- A new integrated assessment tool for improved sediment management is developed.
- Sediment metal recovery could be economically and environmentally beneficial.
- Environmental management impacts are site-specific and depend on time perspective.

GRAPHICAL ABSTRACT



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ABSTRACT

Sediments in ports, marinas and waterways around the world are often contaminated with metals arising from anthropogenic activities. Regular dredging is needed to achieve an appropriate water depth and reduce the environmental impact of pollutants. The aim of this study was to develop an integrated assessment method for comparing various management strategies for dredged sediments at six case study sites in Sweden. Short- and long-term environmental impacts were investigated for different management approaches, including landfilling, deep-sea disposal, metal extraction in combination with the two aforementioned, and natural recovery (no dredging). The potential value of metals in the sediments was estimated using sediment metal contents and current metal prices. Additionally, an assessment of how metal extraction could result in lower management costs was carried out. The cost of the different management approaches was calculated and evaluated together with the corresponding environmental impacts. This study shows that there is a monetary value in dredged materials, in terms of metal content, and that the materials can potentially be used for metal extraction. Metal extraction may also help to reduce the management costs, as cleaner materials are cheaper to handle. The choice of metal recovery method is important in both monetary and environmental terms, potentially contributing to a circular

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economy. In the future, metal recovery may become more profitable, as technologies are improved, and due to probable increases in metal prices and landfill costs.

1. Introduction

Dredging of ports, marinas and waterways must be performed regularly to maintain water depth but can also be performed to remove contaminants (Casper, 2008; Förstner and Apitz, 2007). For example, France, Germany, the Netherlands and United Kingdom dredge around 30–50 M m^3 each annually, while the United States dredges around 200–500 M m^3 (Harrington et al., 2016). In Sweden, 1.4 M tonnes of sediment were dredged in 2016 (SMED, 2018).

Elevated metal concentrations in sediments are often considered a problem, due to their persistence in the sediments and potential negative effects on aquatic organisms (Besser et al., 2018; Jakimska et al., 2011). Metals such as Sn, Cu and Zn, and organotin compounds (OTs) such as tributyltin (TBT) are often found in elevated concentrations in marinas and shipyards, partly due to their use in antifouling paints (Caric et al., 2016; Choi et al., 2014). Tributyltin is toxic even at trace concentrations and has therefore been banned in the EU-15 since 2003 (2002/62/EC), but TBT and its degradation products are still present in sediments and pose a threat to marine ecosystems (Amara et al., 2018; Caric et al., 2016; Egardt et al., 2017; Filipkowska et al., 2014; HELCOM, 2009).

The handling of dredged sediments depends on the level of contamination and local regulations, as specific restrictions, costs and treatment requirements may apply (Casper, 2008). The most common management practices for dredged sediments that cannot be reused are landfilling and deep-sea disposal (Akcil et al., 2015; Bortone et al., 2004), however increasingly stricter environmental legislation affects both the availability of landfilling space and the cost of utilizing it. One example is the European Landfill Directive (1999/31/EC), which has caused some landfills to close, resulting in a reduction in the space available for landfill, and higher landfilling costs (European Environment Agency, 2009). Consequently, stakeholders involved in dredged sediment management are increasingly motivated to investigate alternatives to landfilling. Stabilization and solidification methods, which enable the use of dredged materials in construction, is a common strategy internationally (Mulligan et al., 2001), however its use is limited in Sweden, due to geological conditions (soft clays), high salinity and limited knowledge. Stabilization may be more sustainable if it is combined with a method where metals are extracted from the sediment before it is stabilized. The extraction of metals from sediment can be performed with biological, physical or chemical techniques, which can either be used independently or combined (Akcil et al., 2015; Mulligan et al., 2001). Apart from the production of cleaner sediment, another potential benefit of metal extraction is the opportunity to recover valuable metals, thereby reducing the need for mining. Once metals have been extracted from the sediment, the environmental risks of residual sediments are reduced, and management criteria are more likely to be met.

All available management approaches for dredged contaminated sediments impact the environment in different ways; examples include the reduction of land availability caused by landfilling and the emission of greenhouse gases and particles caused by the transportation of dredged materials (Suer and Andersson-Sköld, 2011; Suer et al., 2009). Integrated assessment (IA) methods, such as multi-criteria analysis, are frequently used to assess multiple conflicting criteria as part of a decision-making process. Integrated assessment has been applied to a wide range of issues, often looking at costs in relation to technical performance and/or impacts on the environment (Renn, 2005; Volchko et al., 2014). The outcomes from this type of method illustrate the impacts on different societal goals and interests (Barnett and O'Neill, 2010; Nyberg et al., 2014).

The study presented here aims at developing and evaluating a new systematic stepwise IA method for assessing the impacts of different management approaches for dredged contaminated sediments. This was achieved through the following objectives: (1) development of a stepwise IA framework; (2) selection of relevant management alternatives for dredged sediments; (3) application of the developed IA framework to six case study areas in Sweden, including collection of relevant data (sediment characteristics, management costs and potential revenues) and the use of the IA framework to compare management alternatives for dredged sediments.

2. Method

The expanded IA method to be developed is generic and considers monetary costs and potential revenues of different management approaches for dredged sediments and the potential environmental impacts of different management approaches for dredged sediments. Here applied on sediment contaminated with metal and TBT at six case study sites in Sweden. The method builds on an IA methodology developed for assessing the environmental impacts of measures to reduce climate-related risks (Andersson-Sköld et al., 2016; Andersson-Sköld et al., 2014), which has here been expanded to include impacts relevant to dredged and contaminated sediment management.

2.1. Case study areas

Six case study areas in Sweden were selected for this study: two ports (P1: Gothenburg; P2 Oskarshamn), three marinas (M1: Björlanda Kile småbåtshamn; M2: Havdens båtklubb; M3: Stenungsunds båtklubb), and one waterway leading into a marina (W: Lövstaviken), see Fig. 1. All sites except P2 are located on the Swedish west coast, where the water has a higher salinity compared to site P2. The sites were selected because they represent a range of activities but also on the data available and the potential to perform additional sampling. General information about each site is provided in Table 1.

2.1.1. Port of Gothenburg (P1)

P1 was established in the 1620s, when the city of Gothenburg was founded in the estuary of the Göta River and is now the largest port in Scandinavia (Göteborgs hamn, 2013). The port has been expanded since its establishment and dredged to allow for larger ships. It is currently dredged every third – fifth year, to retain the acquired water depth. Activities such as storage of various types of cargo and intensive shipyard industries have led to contamination of waters and sediments. Upstream pollution sources, including factories, road traffic, towns, and wastewater treatment plants further contribute to pollutant loads in the sediments of P1. The port and the river are trafficked by recreational boats, cargo vessels, and public transport boats, which contribute boat-associated pollutants, such as Cu, Zn, and TBT (Choi et al., 2014).

Results for P1 will be presented for the overall area, the inner and outer parts of the port as delimited by the border between the river and the estuary.

2.1.2. Port of Oskarshamn (P2)

P2 is located in the southeast of Sweden and dates back to the 19th century. Shipyards were active in the port in the 1860–1960s. Pollutants have also been received from surrounding industries since the middle of

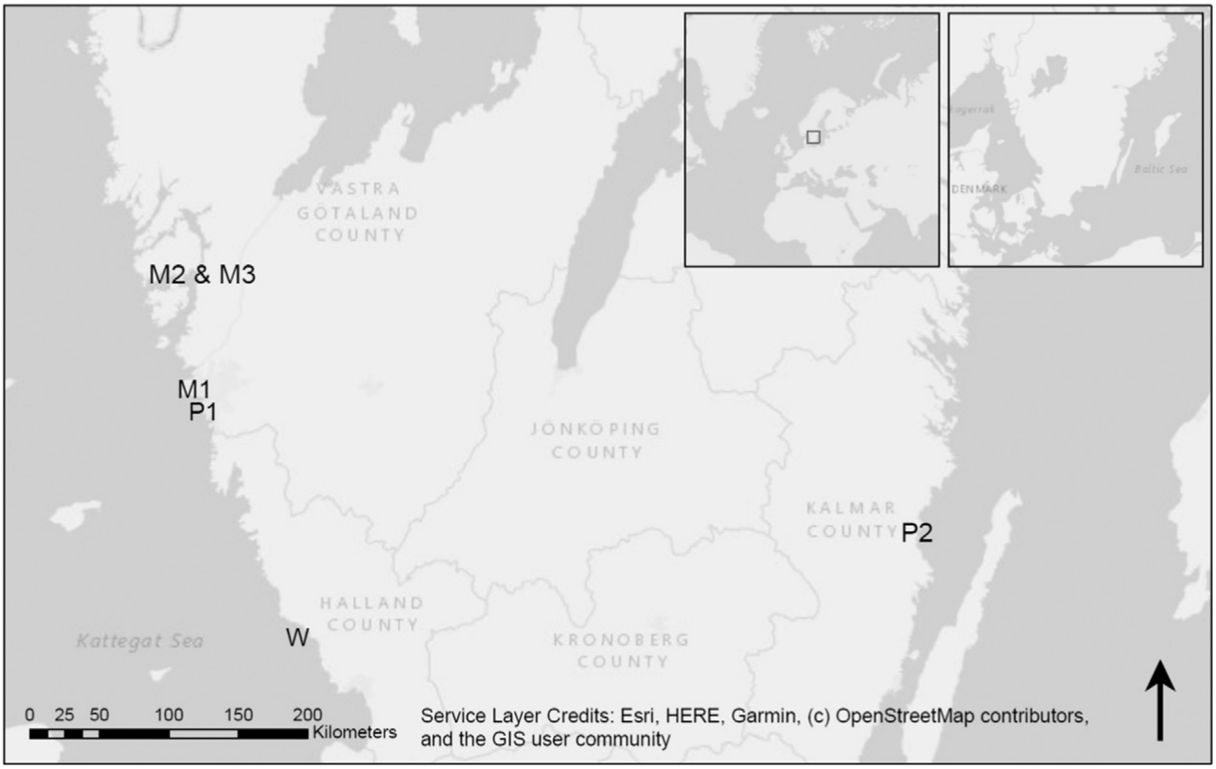

Fig. 1. Location of the case study areas; Port of Gothenburg (P1), Port of Oskarshamn (P2), the marinas Björlanda Kile småbåtshamn (M1), Havdens båtklubb (M2), Stenungsunds båtklubb (M3), and the waterway Lövstaviken (W).

the 19th century; from activities including Cu production, wastewater treatment, and battery production. Dredging of the port was carried out in 2016–2018, to remediate the contaminated sediment. The dredged materials were classified as hazardous waste due to high levels of heavy metals (e.g. Cd) and organic compounds (e.g. dioxins and PCBs) (VBB Viak, 1996) and sent to landfill for hazardous waste. Analysis has been performed on pre-dredging data.

2.1.3. Marinas (M1–M3)

Björlanda Kile småbåtshamn (M1), with 2400 berths, is the largest marina for leisure boats in northern Europe, while M2 and M3 are smaller (Table 1). Stenungsunds båtklubb (M3) is also a marina for leisure boats, while Havdens båtklubb (M2) is a dedicated to boats belonging to employees of a nearby industry (Zeffer and Samuelsson, 2011). All three are located near roads, which potentially contribute traffic-related pollutants to the sediment. All the marinas were active before the use of antifouling paints containing TBT was banned for small vessels, <25 m, in 1989, and for all leisure boats in 1993. Anti-fouling related pollutants as well as other pollutants associated with boating are likely to be found in the sediment.

2.1.4. Lövstaviken waterway (W)

Lövstaviken (W) is a 550 m long waterway, surrounded by quays constructed in the 1950s using landfill material. It is located by the river Ätran and outside the Port of Falkenberg, which has been in use since the 1800s. W was originally built to be used by the nearby Port of Falkenberg but is currently used as a waterway into the marina Lövstavikens båtförening. Possible sources of pollution include leaching from the quay construction and boat-associated pollutants, but also pollutants from current and historic activities upstream.

2.2. Assessment framework

The IA method applied in this study is summarized in Fig. 2 and is further discussed in Steps 1–6 in the text below.

2.2.1. Step 1. Sediment characteristics - concentrations in relation to guideline values at the case study sites

In this step, a pollution/toxicity-based characterization of the sediment is done as basis for identifying potential management strategies. Contaminated sediments are a worldwide environmental concern, but

Table 1
General information about the studied sites, including type of activity, when the site was established, number of berths of the marinas, the size of the site, number of sample locations, and total number of samples.

Site	Name	Activity	Start	Area [m^2]	No of berths	No. of sampling locations	No. of samples	Ref.
P1	Port of Gothenburg	Port	1620s	11,274,000	–	26	52	(COWI, n.d.)
P2	Port of Oskarshamn	Port	1860s	874,500	–	40	103	(VBB Viak, 1996)
M1	Björlanda Kile småbåtshamn	Marina	1971	195,300	2400	3	5	(Björlanda Kile Segelsällskap, n.d.; Sveriges geologiska undersökning, n.d.)
M2	Havdens båtklubb	Marina	~1988	7500	80	2	3	(Zeffer and Samuelsson, 2011)
M3	Stenungsunds båtklubb	Marina	1957	6000	130	2	4	(Zeffer and Samuelsson, 2011)
W	Lövstaviken	Waterway	~1964	37,900	260	6	11	(COWI, 2014)

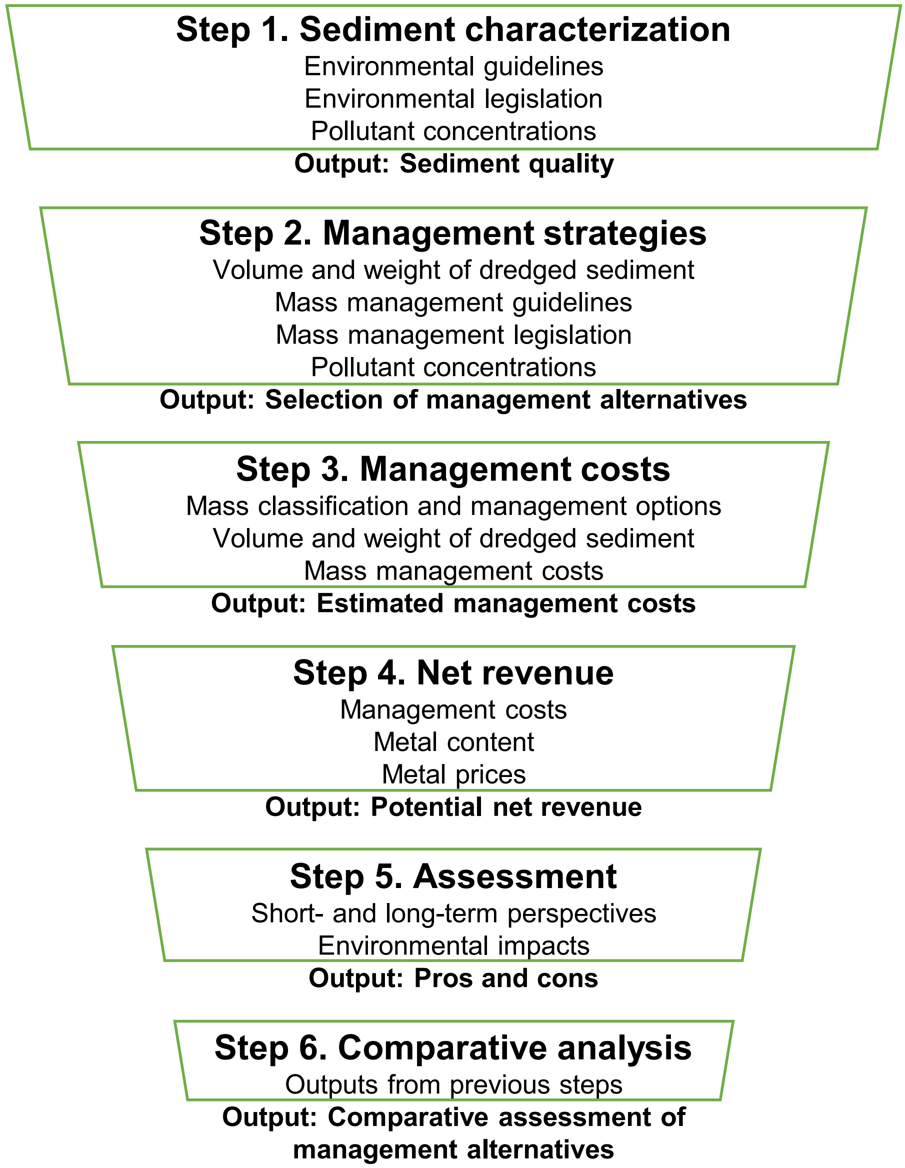

Fig. 2. Assessment framework for the developed integrated assessment method comparing different management approaches for dredged contaminated sediments.

there is no worldwide consensus on the level of pollutants required for sediments to be classified as contaminated, nor for the selection of a management approach (Jersak et al., 2016). Hence, sediment pollutant concentrations must be evaluated with the current environmental and mass management guidelines and criteria for the site of interest.

In Sweden, sediment reference values for metals are only available for limnic sediment (Naturvårdsverket, 2018). Instead, Swedish stakeholders often use Canadian and Norwegian guidelines when assessing the environmental impacts of metals and TBT in marine sediments (INSURE, 2017), as these countries have similar geological conditions to Sweden. Both guidelines have been used in this comparison.

The Canadian Guidelines for Protection of Aquatic Life consist of the Interim Sediment Quality Guidelines (ISQGs) and the Probable Effect Levels (PELs), which are used to evaluate the biological effects of a contaminant (Canadian Council of Ministers of the Environment, n.d.). Concentrations exceeding the PELs frequently result in adverse effects on biota, while levels between the PEL and the ISQG are associated with infrequently occurring adverse effects. Levels below the ISQG rarely cause adverse effects. In Norway, sediments are assigned a class (I-V), based on measured pollutant levels, to indicate their toxicity status. The classes span from Class I (no toxic effects) up to Class V (extensive toxic effects) (Direktoratsgruppen vanndirektivet, 2018; Miljødirektoratet, 2016). The two higher classes, Class IV and V, are most often used for comparison purposes. Currently, there are no toxicity guidelines for sediment in Sweden, which is why no such comparison has been performed. However, toxicity tests using solid and liquid fractions of dredged sediment are recommended to avoid underestimating the risk of disposed sediment for the receiving biota (Rodriguez-Romero et al., 2016; Khosrovyan et al., 2015). Guidelines for hazardous waste (HW) are used to classify dredged materials in Sweden (Avfall Sverige, 2019). If classified as non-hazardous waste, soil quality guidelines are applied to estimate the level of contamination, and to give an indication of appropriate management options. There are two classes of soil guidelines for the protection of environment and human health: 1) the soil guideline for sensitive land use (SLU), which applies to e.g. residential land use; and 2) the soil guideline for less sensitive land use (LSLU), which applies to e.g. industrial land use (Naturvårdsverket, 2009, 2016).

No general guidelines exist for deep-sea disposal in Sweden. Instead, site-specific legal decisions are made for each case, as with the disposal site SSV Vinga (Vinga) on the Swedish west coast that is used as guideline in this study (Svea hovrätt, 2015).

All data used for characterizing the sediment in this study were obtained from previous studies. In addition, complementary samples were taken from M1 and P1. The M1 samples were taken at a depth of 0–10 cm and analyzed for metals and organotins (OTs) using SS EN ISO 17294-1, 2 (mod), EPA-method 200.8 (mod), and ISO 23161:2011. The samples from P1 were analyzed for dry substance and loss on ignition using SS-EN 028113. The concentrations of Cd, Cr, Cu, Hg, Ni, Pb, and Zn were analyzed at all sites, while Ag, As, Ba, Co, Cr (VI), Fe, Mn, Mo, Sb, and V were only measured at some of the sites.

The level of contamination is provided as a range, estimated using the ratio between measured mean and maximum metal concentrations, and the relevant guidelines or criteria. Organotin concentrations are evaluated using the same methodology.

2.2.2. Step 2. Selection of management strategies

In this step, potentially relevant management strategies are identified based on the sediment classification (Step 1) for further assessment. Relevant management strategies could differ nationally but also locally due to e.g., regulation, sediment properties and level of contamination. The four management alternatives used in this study were chosen based on a literature review of three commonly used methods, and metal recovery, Fig. 3.

For the comparative analysis (Step 6 in the IA method) a reference alternative that other alternatives are compared and related to, must be selected. Due to its common practice landfilling was chosen in this study but other reference alternatives as well as management approaches could be implemented in the developed IA method.

Internationally, the most common management approaches are landfill (on land) and deep-sea disposal (Akcil et al., 2015; Bortone et al., 2004), while in Sweden deep-sea disposal is the most common followed by landfilling (417,000 and 226,000 tonnes in 2016) (SMED, 2018).

An alternative to direct landfilling and deep-sea disposal would be to reduce the contaminant concentration in masses by extracting them. This may provide revenue and is potentially a more sustainable management strategy than landfilling and deep-sea disposal (Fathollahzadeh et al., 2012). As the potential for metal extraction increases with increasing contamination levels it is not beneficial to use this strategy on all sediments. Instead, this can be combined with other alternatives such as landfilling. The combined management approaches including metal extraction evaluated in this study are as follows:

1. Materials that meet the criteria for the SSV Vinga (Vinga) are deep-sea disposed, materials that do not meet the Vinga disposal criteria, but are less contaminated than soils suitable for LSLU are landfilled, and the remaining materials are used for metal extraction;
2. Materials that meet the criteria for Vinga are deep-sea disposed, materials that do not meet the Vinga disposal criteria, but are less contaminated than soils suitable for LSLU are landfilled, and the remaining materials are used for metal extraction;
3. Materials that meet the criteria for Vinga are deep-sea disposed, all other materials are used for metal extraction;
4. All materials not classified as hazardous waste (HW) are landfilled, remaining materials are used for metal extraction;
5. Only materials with low-moderate contamination levels (<LSLU) are landfilled, remaining materials are used for metal extraction.

The solid residues after metal extraction is assumed to meet either criteria for deep-sea disposal or <LSLU depending on the combined management approach chosen.

If dredging is not needed to maintain or deepen the current water depth, natural recovery (with or without monitoring) is another management strategy allowing natural processes to degrade contaminants such as TBT to a less toxic form but also include over-sedimentation (Magar and Wenning, 2006). This is mainly an option for the marinas where dredging is mainly done to reduce the pollutant levels but not for ports that need regular dredging.

Stabilization of sediment has not been investigated due to lack of applicability based on the type of sediments in this study (fine grained sediments and high salinity), environmental risk data (for the studied sites) and wider impacts. In-situ treatment, such as capping has not been included as all treatment except (monitored) natural recovery approaches include dredging.

2.2.3. Step 3. Estimation of management costs

Depending on what management strategies are chosen in Step 2, different costs would be applied. Cost for different options varies

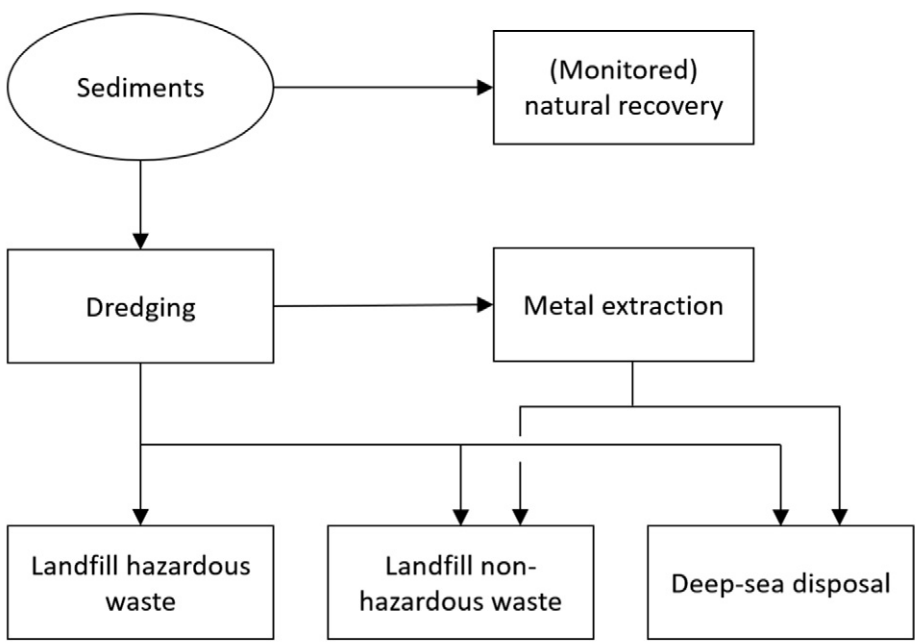

Fig. 3. The potential management approaches for polluted sediments that have been investigated.

internationally and locally. Not only does the mass management itself cost, but other non-direct costs could also occur.

In Sweden, landfilling of sediments is only performed by a small number of stakeholders, and landfill cost are therefore difficult to find, so costs for landfilling of soils have been used. There are large variations in landfilling costs, depending on the location of the landfill and the classification of the waste (Supplementary information Table A (Table A)). The costs for deep-sea disposal have been set to ~2.3 USD/m^3, based on cost estimations by the Water Information Systems Sweden (2012). Mean values have been used in the management cost estimations. Costs associated with transportation of the dredged materials are not included in the estimates, nor are landfill maintenance costs, such as leachate treatment, personnel costs etc.

2.2.4. Step 4. Estimation of net revenue

The dredged sediment can provide both direct and indirect revenue that can be taken into account in this step. Examples of indirect revenue are higher income yielded from larger shipping vessels entering the ports, or revenue from allowing bigger yachts in marinas. Direct revenue can be due to metal recovery from the contaminated sediments. The net revenue is the direct and indirect revenues mines the cists estimated in Step 3. Here, the maximum potential net revenue from metal recovery has been calculated by comparing the cost for different mass management options (Step 3) with the monetary value of the metal content of the sediments (Step 4), added to the savings resulting in relation to the reference alternative.

The metal content was calculated according to the method in Supplementary information B. Here the metal content at the site was used to calculate the potential monetary value, based on available and applicable metal prices. Despite that OTs are problematic in sediments Sn is rarely analyzed. The concentration of tin was calculated based on the proportional contribution of tin to the molecular weight of individual OTs, corresponding to ~41% of the TBT weight, ~51% of DBT, and ~68% of MBT. This was done for all sites except for P2, where no OTs were analyzed. The critical metal extraction cost was also calculated, i.e. the point where the cost of metal extraction equals the potential net revenue.

In this study the 10-year average prices for metals traded at the London Metal Exchange (Table 2) was used for estimating the potential sediment metal value. The annual average values were found to be stable, although with an increasing trend in recent years, especially for Co, Fe and Zn.

2.2.5. Step 5. Assessment of environmental impacts in short- and long-term perspective

All management approaches have other additional environmental impacts, apart from those captured by the toxicity-based guideline values (Step 1). These are discussed in this paper, and semi-quantitative estimations are provided for energy consumption, greenhouse gas emissions (GHG), other emissions to air such as NO_x, particulate matter (PM2.5, PM10), soil or water due to excavation, transport and landfill management of the materials, impacts on land-use caused by landfilling preventing alternative use of a site, and impacts on terrestrial biota/human health (Andersson-Sköld et al., 2016; Andersson-Sköld et al., 2014). Impacts on marine morphology and on marine organisms where deep-sea disposal is part of the management strategies should also be considered. Here, a generally applicable method has been developed to semi-quantitatively estimate impacts based on previous studies and literature and the authors' own judgements based on this information. Generic knowledge, information and experiences are important and useful, but case specific considerations must be taken regarding short-term and long-term perspectives. Short-term perspective applies to the current situation and remediation activity, while the long-term perspective applies when a measure, e.g. a landfill, has been in use for a decade or more.

2.2.6. Step 6. Comparative analysis

In this step the results from Steps 1–5 above should be applied in a comparative analysis based on estimated impact values for acceptable management approaches. The method suggests a relative comparison approach, i.e. the investigated management strategies to be compared with a reference alternative. Short- and long-term environmental impacts should be compared and discussed. All impacts contributing to a positive response (e.g. less land-use or lower emissions than the reference alternative), are given positive values (scoring 1–3), while unwanted effects were given negative values (scoring −3 to −1), no impact equals zero (0). The estimated impacts are calculated to illustrate the relations between the management strategies at each site, by applying the impact values as factors in an equation for calculating relative impact (1).

$$E_j = \sum F_{i,k} * W_k \tag{1}$$

E_j is the estimated total impact for each management approach, *j*

W_k is the relative amount of materials used for the different treatment options, *k*, i.e. deep-sea disposal, landfilling, metal extraction etc. used by each management approach, *j*. Total material amount at each site = 100%

$F_{i,\,k}$ is the impact factor for each impact, *i*, and treatment method, *k*.

3. Results and discussion

The results from each step is presented below and are compiled and compared under Step 6 (Comparison analysis) followed by a short evaluation of the developed method.

3.1. Step 1. Sediment characteristics - concentrations in relation to guideline values at the case study sites

Metal concentrations at the studied sites were characterized by large variations. Table 3 shows the mean concentrations of the most commonly measured pollutants at the case study sites (see Supplementary information Table C (Table C) for full table). The standard deviation (STD) for metal concentrations was of the same magnitude as the mean value at P1, P2 and W, and sometimes higher (Cd, Pb, Hg at P1 and Co, Cu, Mo, Pb, Zn at P2), indicating heterogeneity in the samples. In the marinas, high STDs were found for the OTs, whereas lower levels were found for the metals. In a comparison with metal concentrations in marine sediments at 52 sites worldwide (Qian et al., 2015), most metal concentrations at the case study sites (Table 3) were within the range presented in the article. However, the mean concentration of Cu in P2 exceeded the mean concentrations at all measured sites. For Ni, Pb,

Table 2
Metal prices used in the calculations (London Metal Exchange, 2018) (USD/tonne)[a]. Mean and max over the last 10 years, where this data was available, except for Mo where 2, Ag 3, Fe 4 and Co 9 years of data were available, respectively.

Metal	Mean price (USD/tonne)	Max price (USD/tonne)
Ag	510,000	580,000
Co	39,000	94,000
Cu	6600	10,000
Fe	290	380
Mo	26,000	28,000
Ni	16,000	29,000
Pb	2000	2900
Sn	20,000	33,000
Zn	2200	3500

[a] Bid prices were used for all metals, except Ag and Fe for which sell prices were used, as bid prices were not available. The Fe in the sediment was directly compared to steel scrap prices, as steel scrap was assumed to have the most similar value to the Fe that can be extracted from the sediments, despite that steel also includes other elements, and carries a cost for steel manufacturing.

Table 3
Mean concentration and standard deviation (STD) of the most commonly measured contaminants at the case study sites (STD given for 95% confidence interval).

Compound	P1	P1 inner	P1 outer	P2	M1	M2	M3	W
μg/kg DS								
TBT	150 ± *230*	210 ± *260*	60 ± *110*	n.d.	310 ± *240*	50 ± *50*	210 ± *230*	70 ± *60*
mg/kg DS								
Cd	0.4 ± *0.5*	0.5 ± *0.6*	0.3 ± *0.4*	4 ± *7*	0.2 ± *0.1*	0.4 ± *0.1*	0.2 ± *0*	0.8 ± *0.4*
Cr	40 ± *10*	40 ± *10*	40 ± *10*	50 ± *20*	60 ± *10*	30 ± *0*	30 ± *10*	50 ± *20*
Cu	50 ± *30*	50 ± *30*	40 ± *20*	1100 ± *1400*	190 ± *80*	40 ± *20*	40 ± *20*	40 ± *20*
Ni	20 ± *10*	20 ± *10*	30 ± *10*	60 ± *50*	30 ± *10*	20 ± *0*	20 ± *0*	20 ± *10*
Pb	40 ± *50*	50 ± *60*	30 ± *20*	560 ± *600*	40 ± *10*	20 ± *10*	20 ± *0*	30 ± *10*
Zn	200 ± *100*	200 ± *100*	100 ± *100*	2200 ± *2600*	400 ± *200*	100 ± *100*	100 ± *0*	200 ± *100*

The standard deviation is written in italics.

and Zn in P2 the results were also in the upper range of the measured concentrations. The same was true for Cu and Zn at M1.

The sampling was performed over large areas at each port, within which several current and previous sources of contaminants were present at different locations. P1 also has several sources of contaminants from present and previous upstream activities. In the marinas, the sources depended on for instance traffic and number of berths, while the sources for the waterway included traffic, the marina and the nearby port. In addition to these sources, natural and anthropogenic activities, such as dredging, erosion and sedimentation may change the locations of sediments both horizontally and laterally.

High TBT content in M1 indicated a high usage of boat paint containing TBT. However, the use of this type of paint has decreased, as evidenced by the lower concentration of TBT found in the surface layer compared to the deeper layers (Supplementary information Table D (Table D)). It should be noted that there is a possibility that non-measured antifouling compounds, other than the OTs addressed in this study, such as triphenyltin and its degradation products, have a large impact on the environmental status, which is not shown here (Lagerström et al., 2017).

Comparing the inner and outer parts of P1 in Table C, it was noted that the concentrations in the inner area were higher for all OTs and for 7 (of the 12) metals. However, the concentrations in the inner and outer part were similar, and the deviations were within the standard deviation. The standard deviations were lower for all samples from the outer part, indicating a more homogeneous level of pollution, due to more well-mixed deposition and transported sediments compared to the inner part. The inner area has a longer history of industrial and boat-related activities and is more likely to be affected by upstream activities, which in time may also reach the outer part of the port.

In the inner part of P1, all metal concentrations were higher at the 0.2–0.5 m depth (Table D) than at the surface (0.0–0.2 m depth). At P2, the concentrations either remained the same or rose as the depth increased, except for Hg, for which the concentration was highest at a medium depth, although the variation with depth was within the standard deviation (Tables C–D). The waterway showed a tendency towards higher concentrations of all metals (apart from Cu) and OTs at 0.2–0.5 m depth compared to nearer the surface (Table D). The higher level of contamination deeper down in the sediment indicates that older sources dominate the contamination and that if left untouched, it may be trapped by natural sedimentation patterns. However, the problem will remain, and contaminants could potentially be released during anthropogenic activities such as dredging and boating, but also during natural events, such as storms and land or sediment slides (Peng et al., 2009; Fathollahzadeh et al., 2015). There were few samples taken at the marinas, and where measurements were carried out for different depths, the standard deviation was larger than the deviations between the different depths (Tables C–D).

Table 4 shows the ratios between the mean concentrations and the guideline values for the most commonly measured pollutants at the case study sites. Supplementary information Table E (Table E) presents the mean and max concentrations for metals and OT. It also shows the ratio between measured concentration and guideline value applied in this study including the Vinga disposal criteria.

At site M1, the mean and max values for TBT exceeded all the available guideline values, i.e. the Norwegian, Swedish, and Vinga disposal criteria (Tables 4 and E). The mean and max concentrations of TBT at all the locations for which they were analyzed would be classified as "extensive acute toxic effects for water living organisms" according to the Norwegian sediment toxicity classification. The values in the guidelines were exceeded thousandfold.

As shown in Tables 4 and E, the concentrations of Cu, Pb and Zn were high; as all mean concentrations exceeded the Canadian ISQG at P1, P2, M1, and W. They also exceeded all/several other guideline values at sites P1 and P2. At site W, the mean concentrations of As and Cd, as well as the maximum concentration of Cr, also exceeded both the Canadian ISGQ and the Swedish guidelines for sensitive land use. At M2 and M3, the Cu mean and Zn max, for M3 also As max, concentrations exceeded the Canadian ISQG. The high levels of OTs, and of the metals Cu and Zn, indicate that the marinas are mainly affected by pollutants arising from boating activities.

In summary, all the investigated sediments were contaminated with OTs and/or metals, with mean values exceeding the Norwegian class V (extensive toxic effect). At certain sites (P2, M1), some of the mean metal concentrations exceeded all guideline values (Table E) whereas only one or a few guideline values were exceeded at other sites. The mean concentrations were frequently equal to, or of the same magnitude as, the standard deviation, which indicates a high heterogeneity. This means that the environmental and contamination levels of the sediment were not consistent throughout the site (Tables C–D).

3.2. Step 2. Selection of management approach

Due to variations in contamination within the sites as discussed in Step 1 the sediments should be handled differently. The volume distribution according to the different management approaches based on contamination level is given in Table 5. Most sites have approaches were no metal recovery is performed, but for P2 metal recovery is of interest for all approaches due to the high metal content found in the port. Case study M1 differs from the other sites, as no sediment can be disposed at deep-sea, resulting in a metal recovery performed for all masses for approach 3.

3.3. Step 3. Estimation of management costs

3.3.1. Landfill

In Sweden the cost for landfilling varies depending on contaminant classification, type of landfill and the costs for dewatering (Table A), which gives a total potential landfill (reference alternative) cost for the investigated sites of 0.02 (M2)–739 (P1) million USD (Table 6). This includes only the landfill and dewatering costs, and no additional costs such as transport and leachate treatment.

Table 4
The ratio between the mean concentration and the guideline values for the most commonly measured pollutants at the case study sites. The abbreviations refer to the guideline for which the highest ratio is achieved: ISQG (Canadian ISQG), PEL (Canadian PEL), Class IV (Norwegian class IV), Class V (Norwegian class V), SLU (Swedish class SLU), LSLU (Swedish class LSLU), and Vinga (SSV Vinga criteria).

Compound	P1	P2	M1	M2	M3	W
TBT	9400 (Class IV) 4700 (Class V) 1.0 (SLU) 3.0 (Vinga)	n.a.	19,000 (Class IV) 9600 (Class V) 2.1 (SLU) 1.0 (LSLU) 6.2 (Vinga)	3000 (Class IV) 1500 (Class V)	13,000 (Class IV) 6600 (Class V) 1.4 (SLU) 4.2 (Vinga)	4300 (Class IV) 2100 (Class V) 1.4 (Vinga)
Cd	–	5.6 (ISQG) 4.9 (SLU) 1.3 (Vinga)	1.2 (ISQG)	–	–	1.2 (ISQG) 1.0 (SLU)
Cr	–	–	–	–	–	–
Cu	2.4 (ISQG)	58 (ISQG) 10 (PEL) 13 (Class IV) 7.4 (Class V) 14 (SLU) 5.4 (LSLU) 14 (Vinga)	10 (ISQG) 1.8 (PEL) 2.3 (Class IV) 1.3 (Class V) 2.4 (SLU) 2.4 (Vinga)	1.9 (ISQG)	2.1 (ISQG)	2.3 (ISQG)
Ni	–	1.4 (SLU)	–	–	–	–
Pb	1.4 (ISQG)	19 (ISQG) 5.0 (PEL) 11 (SLU) 1.4 (LSLU) 5.1 (Vinga)	1.4 (ISQG)	–	–	1.1 (ISQG)
Zn	1.3 (ISQG)	18 (ISQG) 8.2 (PEL) 3.0 (Class IV) 8.9 (SLU) 4.4 (LSLU) 6.2 (Vinga)	3.5 (ISQG) 1.6 (PEL) 1.7 (SLU) 1.2 (Vinga)	–	–	1.3 (ISQG)

3.3.2. Deep-sea disposal

At all sites, apart for M2, the mean metal and/or TBT concentrations exceeded the permitted limits for disposal at the Swedish deep-sea disposal site SSV Vinga (Vinga) (Table E). At M2, only the TBT concentration exceeded the upper limit for disposal at Vinga. However, in all samples from M1 at least one parameter exceeded the deep-sea disposal limit. Apart from M1, all sites had sample points where the concentrations of all measured contaminants met the criteria for deep-sea disposal. For these materials, deep-sea disposal would be possible, as exemplified for P1 in Supplementary information Table F (Table F) (sample points marked with "OK" in the Vinga column). For P1, these materials made up approximately 52% of the total volume investigated. The corresponding volumes at P2, M1, M2, M3, and W made up 19%, 0%, 77%, 50% and 57%, respectively, based on the concentrations at the individual sampling points. The management costs for deep-sea disposal of the proportion of the materials for which this would be permitted would range from 0 (M1)–6.7 million USD (P1) (Table 6), based on the contamination level, the total volume that meets the criteria for this type of disposal, and the cost for deep-sea disposal.

The natural recovery approach has no direct costs, however, monitorial sampling and evaluation would require some financing (a relatively low cost, therefore not included here).

Table 5
Sediment volumes (%) classed according to mass management criteria (landfill only or a combination of landfill and deep-sea disposal).

Sediment volumes [%]	P1	P2	M1	M2	M3	W
Approach 1						
Metal extraction	0	31	0	0	0	0
Deep-sea disposal	52	19	0	77	50	57
Landfill	48	49	100	23	50	43
Approach 2						
Metal extraction	16	66	40	0	50	0
Deep-sea disposal	52	19	0	77	50	57
Landfill	32	15	60	23	0	43
Approach 3						
Metal extraction	48	81	100	23	50	43
Deep-sea disposal	52	19	0	77	50	57
Landfill	–	–	–	–	–	–
Approach 4						
Metal extraction	0	31	0	0	0	0
Deep-sea disposal	–	–	–	–	–	–
Landfill	100	69	100	100	100	100
Approach 5						
Metal extraction	16	66	40	0	50	0
Deep-sea disposal	–	–	–	–	–	–
Landfill	84	34	60	100	50	100

3.4. Step 4. Estimation of net revenue

For P1, the total potential value of the measured metals was 4.7 million USD (Table F). Of the metals, Co and Ni contributed 35% each, and Cu provided 23% of the total value. For treatment alternative 1–5, the potential metal value varied from zero (approaches 1 and 4), in cases where no parts of the materials were classified as HW, i.e. no metal recovery, to 1 million USD for approaches 2 and 5, and 2.5 million USD for approach 3. The potential value of the metals was low compared to the potential net revenue; 0.2–2% for approaches 2 and 4, and 0.3–4% for approach 3 (Table 7). Where this is the case, a more attractive option may be to focus mainly on extraction of the steering OTs and metals (i.e. TBT and As at many sampling spots, and Ba, Hg and Zn at some others, Tables 4 and F), depending on the cost, effectiveness and availability of extraction methods for these metals, compared to the more valuable ones. The net revenue, estimated as the sum of the potential metal value, saved landfill costs and final treatment costs (here the same costs as for deep-sea disposal), at P1 varied between 15 and 730 million USD (Table 7), depending on treatment and actual landfill costs (Table 6).

At P2, the estimated total potential metal value was 4.6 million USD (Table F). The Fe content contributed with 59%, Cu with 18%, and Zn

Table 6
Management cost ranges for deep-sea disposal, as permitted, and landfill (USD). Min and max represent the cost when the lowest and highest disposal costs are applied, respectively.

	P1		P2		M1		M2		M3		W	
	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
Landfill only	81,000,000	740,000,000	7,900,000	47,000,000	1,400,000	10,000,000	19,000	400,000	73,000	370,000	130,000	2,500,000
Deep-sea disposal	–	6,700,000	–	190,000	–	–	–	6700	–	3500	–	26,000
Remaining landfill	60,000,000	360,000,000	7,300,000	37,000,000	1,400,000	10,000,000	3300	75,000	65,000	180,000	50,000	1,000,000
Sum deep-sea and landfill	66,000,000	360,000,000	7,500,000	37,000,000	1,400,000	10,000,000	10,000	82,000	68,000	190,000	80,000	1,000,000

with 13%. The saved landfill costs varied from 3.4 million USD (approach 4) to 46 million USD (approach 3). The net potential revenue was highest, 11–51 million USD, for approach 3 (that is materials that meet the criteria for Vinga are deep-sea disposed, all other materials are used for metal extraction). For P2, the estimated potential metal value was of the same magnitude as the saved landfill costs for all approaches, 1–5 (Table 7). Accordingly, the potential net revenue depends on both the metal recovery potential and the landfill cost savings.

For M1, approach 3 would result in all metals being extracted when no parts of the sediment meet the Vinga criteria, resulting in 0.1 million USD and a potential net revenue of 1.3–10 million USD. M2, M3 and W had low metal values and the materials were fairly clean (Tables 4, 7, E–F). The sediments from M2 and M3 were very clean, which means that the estimated net potential revenue was low, and the option of leaving the sediment in place (natural recovery) had the highest potential net revenue of all the approaches. For W, all materials are <LSLU, which gave a neutral net potential revenue for approaches 4 and 5, as all the materials would be landfilled, while approach 3 offered a potential net revenue of 0.1–2.4 million USD (W). All the management approaches, apart from the natural recovery option, would incur costs related to dredging/excavation and transportation, as well as other additional costs, on top of those presented in Table 7.

For P1, M2, and M3, the metal contributing the most to the value was Ni (together with Co). P1 also had high levels of Cu, which would contribute to the total value if extraction was performed. Interestingly, Fe was contributing most to the revenue, despite not being the most valuable metal at P2 (Table F); despite its low price, the large quantities would make it interesting to extract. Fathollahzadeh et al. (2012) have also claimed that Cu and Pb are economically feasible to recover during the dredging of the port. In some scenarios it may be worth extracting metals for the sole purpose of spending less money on landfill. It may be of interest to target specific areas and treat only these.

Increasing metal prices has been a trend for the last 10 years, and the value of the metals in the sediments is likely to increase in the future. The average metal price used in this study produced a lower gain than if metal recovery was to be performed today. Moreover, to what degree the metals should be recovered from the sediment could be discussed. At sites such as P1, where the highest net potential revenue relates to saved landfill costs, perhaps only partial metal recovery would be of interest. For sites such as P2, where the value of the metal contributes to the potential net value, it may be economically beneficial to perform a full metal recovery to extract most metals. However, the cost of extracting and processing the metals into usable forms would probably determine whether this would be carried out in reality or not. To achieve the same total management cost as if all sediments were sent to landfill, the highest acceptable cost of the metal extraction would vary between 120 USD/tonne (P1 approach 5) to 1350 USD/tonne (M2 approach 3) (Table 8). For M2 in particular, current methods would yield a positive net value, with metal recovery included in the net revenue.

Examples of full-scale projects with metal recovery from sediments are limited. Only one full-scale project with metal recovery (Mulligan

Table 7
Potential metal revenue and potential savings from metal extraction compared to landfill of all materials (USD). Bold indicates the option with the highest potential value/saving.

Site	Approach	Potential metal value (all metals)	Saved landfill cost		Potential net revenue[a]	
			Min	Max	Min	Max
P1	1	0	15,000,000	376,000,000	15,000,000	380,000,000
	2	970,000	57,000,000	490,000,000	58,000,000	490,000,000
	3	**2,500,000**	**68,000,000**	**726,000,000**	**71,000,000**	**730,000,000**
	4	0	0	0	0	0
	5	970,000	42,000,000	114,000,000	43,000,000	120,000,000
P2	1	2,700,000	3,700,000	24,000,000	6,400,000	26,000,000
	2	4,000,000	6,500,000	38,000,000	10,000,000	42,000,000
	3	**4,400,000**	**6,900,000**	**46,000,000**	**11,000,000**	**51,000,000**
	4	2,700,000	3,400,000	14,000,000	6,100,000	16,000,000
	5	4,000,000	6,100,000	28,000,000	10,000,000	32,000,000
M1	1 & 4	0	0	0	0	0
	2 & 5	47,000	1,000,000	4,000,000	1,000,000	4,100,000
	3	**100,000**	1,200,000	**10,000,000**	1,300,000	**10,000,000**
	Natural recovery	0	**1,400,000**	**10,000,000**	**1,400,000**	**10,000,000**
M2	1 & 2	0	9000	320,000	9000	320,000
	3	**180**	10,000	390,000	10,000	390,000
	4 & 5	0	0	0	0	0
	Natural recovery	0	**19,000**	**400,000**	**19,000**	**400,000**
M3	1	0	5200	180,000	5200	180,000
	2 & 3	**560**	66,000	360,000	67,000	360,000
	4	0	0	0	0	0
	5	**560**	61,000	180,000	62,000	180,000
	Natural recovery	0	**73,000**	**370,000**	**73,000**	**370,000**
W	1 & 2	0	50,000	1,500,000	50,000	1,500,000
	3	**6200**	**79,000**	**2,400,000**	**85,000**	**2,400,000**
	4 & 5	0	0	0	0	0

[a] Potential metal value and net saved management cost, including costs for deep-sea disposal of remediated materials. Metal extraction cost is not included.

Table 8
Maximum allowed extraction cost (USD/tonne) for the different treatment approaches at the different case study sites to be cost neutral with landfilling all materials, based on the mean landfill cost presented in Table A.

Site	Approach				
	1	2	3	4	5
	USD/tonne				
P1	–	370	170	–	120
P2	390	310	230	280	250
M1	–	170	140	–	170
M2	–	–	1350	–	–
M3	–	210	210	–	130
W	–	–	310	–	–

et al., 2001), was identified in literature, but did not describe further commercialization of the project or investigate environmental impacts. The metal recovery costs based on their full-scale project were 100–250 USD/tonne, which is comparable to the maximum acceptable metal recovery cost for the sites in this study (Table 8). However, as the degree of extraction is likely to vary for different metals, sediments and methods, a direct comparison between different sites is difficult to make. Soft clays, which usually make up the bulk of Swedish sediments, are considerably more difficult to treat than coarser sediment fractions, such as sand. Consequently, treatment of clay-type sediment is probably more expensive. It is difficult to assess how accurately an estimation can be established, both in terms of metal content and recovery possibilities. The predicted metal content is also dependent on sampling and extraction methods. Differences in aim for previous studies meant that not all metals of economic interest were analyzed at all sites and the number of samples varied (Tables 1 and C).

3.5. Step 5. Assessment of environmental impacts from a short- and long-term perspective

Supplementary information Table G (Table G) contains a summary of environmental impacts not related to the guideline values for contaminated sediment, contaminated land or landfill.

3.5.1. Landfill

Landfills require land areas to be occupied thereby preventing the land from being used for other purposes. Both during and after the active period, a landfill may also have negative impacts on local ecosystems and related ecosystem services, including aesthetic value and biodiversity (Camerini and Groppali, 2014; Tribot et al., 2018; Yazdani et al., 2015). Other environmental impacts include use of resources (energy and materials), emissions to air, soil and water, as well as occupational risks related to excavation, transport of the excavated materials, and landfill management (Suer and Andersson-Sköld, 2011). The landfills for hazardous waste are only available at a limited number of locations in Sweden, increasing the likelihood that materials must be transported over long distances. Hazardous waste also demands more rigorous management of the landfill, which in turn uses more energy and water. The landfill leachate water also requires more management than for non-hazardous waste, and inert waste in particular (Andersson-Sköld et al., 2014; Suer and Andersson-Sköld, 2011; Suer et al., 2009). The geotechnical stability of a landfill must also be assessed, to prevent landslides, which could potentially expose contaminated materials. In addition, as sediments have a high water-content, the need for leachate treatment is higher than for ordinary soils. For marine sediments the water is also saline, which further limits the number of suitable landfill sites, as release of saline water is often restricted. Depending on the amount of organic content within the sediment, landfill gas treatment may be required, to avoid release of GHG and the risk of explosion.

The main advantage of landfilling is that the contaminated materials are controlled and collected in the same area. In addition, urban mining from former landfills may offer an opportunity to extract metals and other resources within the not too distant future.

3.5.2. Deep-sea disposal

Important considerations for deep-sea disposal are the geological and hydrologic conditions at the site, which should preferably be an accumulation bottom. In addition, only materials with concentrations below pre-defined, site specific levels are permitted.

There are several negative aspects of deep-sea disposal, in addition to the energy and resource consumption, and potential emissions to air, soil and water caused by excavation and transportation of the materials. If the materials are contaminated, the impacts are related to contamination levels in relation to toxic risks. In addition to the metals and OTs investigated here, materials containing elevated concentrations of compounds such as persistent organic pollutants (POPs) and microplastics should also be considered, due to the severe risks they pose to marine organisms (Auta et al., 2017). Here, however, materials with contaminant concentrations below the SSV Vinga limits are assumed to be acceptable with regard to other contaminants as well, which limits the impacts caused by the contaminants per se. However, other impacts may arise as a result of the depositing of the materials, i.e. physical disturbances. The covering of demersal organisms and changes in the morphology, chemical composition and nature of the bottom substrate may affect the benthic society, and result in increased turbidity and suspended solids, which may affect primary production and the growth of filtering organisms (OSPAR Commission, 2009; Witt et al., 2004). Benthic organisms living on, or near, the seabed can be buried, while digging organisms may be able to move up through the deposited material and fish are able to leave the area (Bortman, 2003). Further, invertebrates ingesting food (such as detritus-feeders, filterers, carnivores/omnivores, etc.) are greatly influenced, while epibenthic mobile crustaceans (e.g. *Neomysis integer*, *Schistomysis kervillei* and *Praunus flexuosus*) may even be more common at a dumping site (Witt et al., 2004). Some organisms, such as seaweeds, coral reefs and oyster banks never recover (Bortman, 2003). Those organisms, however, do not occur naturally at deep-sea sites in Sweden. The area affected by the deposits may extend beyond the dumping site due to suspension and dispersion. The size of the area affected depends on currents, depth, seabed type, as well as the properties of the dumped materials, weather conditions, and the disposal method applied (OSPAR Commission, 2008). Therefore, any site selected for deep-sea disposal must be carefully chosen based on measurements and modellings of site-specific conditions, and the disposal must be carried out under conditions that minimize suspension and dispersion. Long-distance transports should preferably be avoided. If antifouling paint is used on the transport vessel there is risk of pollutant spreading.

The recovery time for biota after sediment disposal varies depending on several factors, including the water streams, but the recovery time for normally undisturbed marine environments is typically 1–4 years (Bolam and Rees, 2003).

A previous study has shown that both the energy consumption and emissions of CO_2 and other greenhouse gases, are significantly lower when materials are transported by a sea vessel (e.g. barge) to when they are transported a similar distance by truck (Andersson-Sköld, 2015; Hammarstrand and Millander, 2015), which is one advantage of deep-sea disposal over most landfill alternatives. However, the emissions to air of nitrogen oxides (NOx) and particles (PM2.5 and PM10) from a common barge can be high (Andersson-Sköld, 2015; Haeger-Eugensson et al., 2015), although this will depend on the type of vessel used. Modern barges equipped with NO_x catalysts, such as SCR (Selective catalytic reduction), and vessels adapted to the sulfur limits according to the Sulfur Emission Control Areas (ECAs) have significantly lower emissions of NO_x, sulfur and particles compared to earlier types of ships and barges (Fridell et al., 2008; Trafikanalys, 2017).

3.5.3. Metal extraction

Metal extraction from contaminated sediments offers an opportunity to recover metals, thereby providing an alternative to juvenile mining that could save energy and other resources, reduce emissions from the mining processes, and minimize the impacts on the landscape and mining environment, potentially contributing to a circular economy. As mining often takes place at locations far from where the metals will be used, and may involve several transport steps during the refining and upgrading process, the distance that contaminated sediments would have to be transported for metal extraction is likely to be shorter, which would lead to savings in both energy and resources, as well as decreased emissions caused by transportation.

Metal extraction is a complex issue and the environmental impacts and potential health risks as well as the extraction efficiency depend on the method applied. The choice of extraction method is due to e.g. the contamination level, the metal speciation and the sediment matrix properties. In addition, the quality of the final sediment residues must be addressed as it determines whether it must be landfilled (preferably as less contaminated at a lower cost), or can be deep-sea disposed (Akcil et al., 2015). Sediment metal recovery techniques with less environmental impact are currently being developed but have not yet been applied in any large-scale dredging projects (Akcil et al., 2015).

However, metal recovery from other contaminated materials, like ashes and soils, is being performed at both laboratory and full scale, and this practice has been shown to be more beneficial than direct landfilling (Karlfeldt Fedje et al., 2014; Schlumberger et al., 2007). In a study on Cu recovery from soil, it was shown that the economic potential is low compared to landfilling, however the environmental impact was reduced (Volchko et al., 2017). In addition, metal price was an important influencing factor on the results, which suggests that increasing metal prices could make recovery more economically attractive than landfilling. However, it should be noted that the amounts of interesting metals are significantly higher in ashes and many soils than in most sediments, which means that the amounts yielded from sediments are likely to be smaller.

As a consequence of the lack of comparable full-scale metal recovery methods for sediments, the environmental impacts nor their magnitude are possible to estimate. However, a positive impact will be reduced land use as virgin metal resources potentially could be saved. Impacts in terms of risk, energy consumption, emissions and air quality will be heavily dependent on the method used but will mainly affect a short-term time frame.

3.5.4. Natural recovery

A natural recovery approach (with or without monitoring) is not feasible for P1, P2, or W, as these sites require frequent dredging. However, it would be an option for the marinas if only the sediment metals and OTs concentrations investigated here were considered. Additional toxicity test results and analyses of other compounds, such as bioaccumulating POPs and microplastics may, however, still motivate remediation actions (Dubaish and Liebezeit, 2013; Frias et al., 2016; Jiang, 2018). The advantages of natural recovery are that this does not lead to any extra energy or resource consumption, nor does it result in emissions to air, soil or water due to excavation, transport, landfill management or metal extraction. Moreover, this approach does not require any additional land (or deep-sea) areas to be set aside for material deposits, and it does not alter the current morphology neither at the site nor for disposal purposes. In addition, it does not cause any new risks to the workers carrying out the activities.

3.6. Step 6. Comparative analysis

Table 9 compares the different investigated management approaches in terms of potential net revenue and the estimated total short- and long-term environmental impacts (other than those related to the guideline values for contaminated sediment, contaminated soil or landfill criteria), normalized to the option of putting all materials into landfill. The impacts for each individual site are provided in Supplementary information Table H (Table H). The estimated impacts have been calculated to illustrate the relation between the management strategies at each site, by applying the impact values provided in Table G as factors for the relative impact applying Eq. (1) presented in Section 2.1.

The monetary costs and potential revenues of the different management approaches for dredged sediments are available in Tables 6 and 9. The net costs and revenue vary from zero to a potential net revenue of 730 million USD, depending on contamination level, total metal content, and management strategy. In most cases, the saving from reduced landfill costs is much greater than the potential metal revenue (Table 7). The exception is P2, where the potential metal recovery revenue is of the

Table 9
A comparison of the potential net revenues and estimated environmental impacts, other than in relation to the guideline values for contaminated sediment, contaminated land or landfill criteria for the different approaches. Both potential net revenue and total environmental impacts are relative to the option of landfilling all materials. This means that the total short-term environmental impact can potentially vary between −10 and 32, and long-term between −16 and 26, where the first numbers represents the least and the second the most favorable impact, while an impact of 0 would be equal to the impact of landfilling.

Site	Approach[a]	Potential net revenue[b] (USD)		Total env. impacts	
		Min	Max	Short-term	Long-term
P1	1	15,000,000	380,000,000	1	2.1
	2	58,000,000	490,000,000	1.4	2.7
	3	**71,000,000**	**730,000,000**	**2**	**4**
	4	0	0	0	0
	5	43,000,000	120,000,000	0.3	0.6
P2	1	6,400,000	26,000,000	1	2
	2	10,000,000	42,000,000	1.5	2.9
	3	**11,000,000**	**51,000,000**	**2**	**4**
	4	6,100,000	16,000,000	0.4	0.9
	5	10,000,000	32,000,000	0.9	1.8
M1	1	0	0	0	0
	2	1,000,000	4,100,000	0.8	1.5
	3	**1,300,000**	**10,000,000**	**2**	**4**
	4	0	0	0	0
	5	1,000,000	4,100,000	0.8	1.5
	Natural recovery	**1,400,000**	**10,000,000**	**10**	2
M2	1	9000	320,000	1.8	3.7
	2	9000	320,000	1.8	3.7
	3	**10,000**	**390,000**	**2**	**4**
	4	0	0	0	0
	5	0	0	0	0
	Natural recovery	**19,000**	**400,000**	**10**	2
M3	1	5200	180,000	1	2
	2	67,000	360,000	**2**	**4**
	3	67,000	360,000	**2**	**4**
	4	0	0	0	0
	5	62,000	180,000	1	2
	Natural recovery	**73,000**	**370,000**	**10**	2
W	1	50,000	1,500,000	1.5	3
	2	50,000	1,500,000	1.5	3
	3	**85,000**	**2,400,000**	**2**	**4**
	4	0	0	0	0
	5	0	0	0	0

The numbers written in bold are the highest value yielded for each case study. For the marinas, the highest value that are not the approach Natural recovery are also written in bold.

[a] Approach 1. Materials that meet the SSV Vinga (Vinga) criteria are deep-sea disposed, materials exceeding the Vinga disposal criteria and < hazardous waste (HW) are landfilled and the remaining materials are used for metal extraction. Approach 2. Materials that meet the Vinga criteria are deep-sea disposed, materials exceeding the Vinga disposal criteria and < LSLU are landfilled, and the remaining materials are used for metal extraction. Approach 3. Materials that meet the Vinga criteria are deep-sea disposed, materials exceeding the Vinga disposal criteria are used for metal extraction. Approach 4. All materials < HW are landfilled and the remaining materials are used for metal extraction. Approach 5. Only materials with low-moderate contamination level (<LSLU) are landfilled and the remaining materials are used for metal extraction.

[b] Potential net revenue includes potential metal value and net saved management cost.

same magnitude as the saving. The option with the highest potential net revenue is approach 3, where all materials that meet the Vinga criteria are deep-sea disposed, and the remaining materials are used for metal extraction. For the marinas, the most monetary beneficial alternative would, however, be natural recovery (with or without monitoring) due to the avoidance of costs for disposal and landfill.

The metal extraction cost required to achieve a net balance between metal monetary gain, saved landfill cost and new disposal cost, varies depending on the choice of management approach and the price of metals. When the mean metal price is used for approach 3, which allows the highest metal extraction cost, this gives 170 USD/tonne for P1, and 230, 140, 1350, 210, and 310 USD/tonne for P2, M1, M2, M3, and W, respectively (Table 8). With the exception of M2 and W, these costs are all within the range of metal recovery cost estimated by Mulligan et al. (2001).

Several potential negative environmental impacts have been identified for all the different management approaches, as summarized in Table G for the main management strategies, and in Tables 9 and H for the investigated site-specific options, ranked compared to the option of landfilling all materials.

Table 9 indicates that approach 3 is the most sustainable option for all sites, as both the monetary and environmental impacts, short- and long-term, are the most favorable (possibly exception for the natural recovery approach for the marinas).

For the marinas, the most favorable option from an economical and short-term environmental perspective, is the natural recovery approach, which provides no recovered metals but saves landfill costs, thereby achieving a high net revenue (Tables 7 and 9). Based on the information in Tables 9 and H the natural recovery approach is the most beneficial short-term management strategy from an environmental perspective. This is because it does not cause any additional environmental impacts. This option may, however, involve a risk to marine organisms from pollutants in the sediment, as dredging may be an effective way to lower that risk. Before any action is taken, it is recommended that further site-specific investigations for other pollutants than those investigated here are carried out (e.g., POPs or microplastics), as well as an assessment of both acute and long-term toxicological risks. However, from a long-term perspective, approach 3 provides the most environmental benefits, which implies that metal recovery may be a feasible option in the future (Table 9).

Whether the natural recovery approach or approach 3 should be selected for the marinas would depend on the importance placed on short-term impacts in relation to longer-term impacts, and on how the individual impacts are valued in relation to each other. This valuation depends on how the different impacts are perceived by the involved stakeholders, as well as the willingness, ability, and requirements to achieve the most preferred option among regulators, decision-makers and stakeholders. The values are context (time and site) dependent and will change based on several factors. As an example, higher pollutant concentrations in the sediments may cause different approaches to become the most beneficial, or more beneficial than they were in this study. This is demonstrated by the fact that the potential net revenue from approach 4 is higher for the most polluted site (P2) than for the other sites investigated here. Concentrations may be higher in other ports and marinas, which may result in higher potential revenues from a greater proportion of the materials than was the case for the sites investigated in this study. Furthermore, changes in the market values of metals may alter the potential revenues, and decisions to value one environmental impact higher than another may give a different result from those obtained here, where no weighting has been applied.

3.7. Applicability of the developed IA method

As illustrated here, an integrated assessment may be used to identify the least beneficial alternatives early in a decision-making process and to determine which options to subject to a more in-depth analysis. The developed IA method presented in this work builds on methods by Andersson-Sköld et al. (2014, 2016) that provided an applicable, useful and pragmatic semi-qualitative basis for land use planning. The result may be that one approach stands out as more positive than the others, both from a monetary and environmental perspective, but there will most likely be conflicts of interests, where short-time environmental impacts have to be weighed against potential revenue and long-time potential environmental impacts and may differ from site to site.

For successful implementation of the preferred approach, irrespective of which option is chosen, such conflicts should be addressed and considered in a transparent way as part of the decision-making process (Andersson-Sköld et al., 2016; Glaas et al., 2010; Storbjörk and Hjerpe, 2013). The choice of management approach may need to be based on a more in-depth analysis of the potential costs, revenues, and environmental impacts. In addition, the views and values of stakeholders and experts must be taken into account, to inform the weighting of the aspects being considered. Weighting the impacts may provide a clearer basis for the decision. This process will also provide a more structured base for communication in the forthcoming implementation process (Andersson-Sköld et al., 2016; Glaas et al., 2010; Storbjörk and Hjerpe, 2013).

In cases where it is not evident which approach would be the most beneficial the integrated assessment results provide information about impacts, and a basis for discussion. It also illustrates conflicting, and occasionally synergetic, effects. This type of assessment would be relevant early in a decision-making process, to identify the options least relevant from an economic and environmental perspective. The social pillar of sustainability is of high relevance, both in relation to transparency and because it needs to be applied when the impacts of the most beneficial options are weighted. The developed method can be applied separately, or as a complement to the recent sediment management model by Harrington et al. (2016), which mainly focuses on economical and socio-economical aspects. Moreover, the aspects considered in this comparison of management approaches can be applied as a checklist of environmental and monetary aspects to consider in decision-making processes for dredged sediment management strategies elsewhere.

To apply the latest market values is of high relevance in the assessment. One finding from this study is that Co was present in the sediment in concentrations of economic interest. Many metals with a monetary value to society are not traditionally investigated as part of a sediment analysis, as they are considered low risk. If metals with a potential monetary value were also to be included in these types of assessments, their economic value may contribute to a higher potential net revenue. Another aspect to consider is that although some metals may have a low monetary value, they may have severe effects on aquatic life. As an example, Cd was present at high levels at site P2, and its toxicity alone motivates a full-scale dredging of the entire port. In sediments with high metal concentration metal recovery may provide an economic and environmental motivation to capture the metals before they are spread further in the environment. Furthermore, in areas where the sediment geotechnical quality is higher than in this study, the value of the sediment should also be included as a potential source of revenue before and after metal recovery. Additionally, when assessing a site from environmental and economic points of view, it may be sufficient to extract selected metals to meet certain environmental, disposal, and/or utilization criteria by lowering the toxicity of the materials and thereby the disposal cost. In other cases, recovering all or some of the metals may be more desirable, if the aim is to attain precious metals.

The studied method will be useful both to identify the most and least relevant management approaches for dredged sediments and, not least, offers transparency and illuminates conflicts of interest early in a decision-making process.

4. Conclusions

- The developed method will be useful, as integrated assessments can identify the most and least relevant management approaches for dredged sediments. It also offers transparency and illuminates conflicts of interest early in a decision-making process.
- It was found that the short-term environmental impacts differ from the potential long-term impacts of different management approaches.
- The net revenue of the different approaches was found to depend both on the potential to reduce management costs and on potential revenue from metal recovery.
- For the investigated ports and waterway, the option of dumping dredged materials classified as clean at a deep-sea disposal site and using the remaining materials for metal recovery was found to yield the highest potential net revenue as well as the lowest short- and long-term environmental impacts of the considered approaches.
- For marinas, the same approach scored the highest long-term environmental impact, while the option to perform natural recovery (with or without monitoring) yielded the highest potential net revenue and most favorable short-term environmental impact.
- The handling costs and choice of metal recovery technique would affect the overall management results; techniques with low environmental impact should be favored have not yet been developed.
- Increasing metal prices, higher landfill costs, and cheaper metal recovery techniques could make metal recovery from sediments a more attractive option and contribute to a circular economy.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary information

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